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Investigative approaches to urban biogeochemical cycles: New York metropolitan area and Baltimore as case studies

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Introduction

By 2007 more than half of the world's population is expected to reside in cities (United Nations, 2004). As urban populations and the number of cities expand, natural and agricultural lands are transformed into highly altered landscapes. These changes in demography and land use have contributed to the alteration of biogeochemical cycles at local, regional and global scales (Vitousek *et al.*, 1997a; Pouyat *et al.*, 2003). Yet we lack sufficient data with which to assess the underlying mechanisms of land-use change (Groffman *et al.*, 2004), largely because of the difficulty encountered when applying established biogeochemical research methods such as large-scale field manipulations to urban and suburban ecosystems (Pouyat *et al.*, 1995a). Moreover, current conceptual and quantitative biogeochemical models incorporate human effects only indirectly (Groffman and Likens, 1994).

As a result, most urban ecosystem studies have relied on a comparative approach or 'natural experiments' to investigate urban effects on biogeochemical cycles in ecological remnants characteristic of a particular area or region (Pickett *et al.*, 2001). This approach takes advantage of remnant systems as 'whole ecosystem' manipulations by which the effects of multiple urban stress and disturbance factors are assessed with established statistical methods and modelling approaches (Pouyat *et al.*, 1995a; Breitburg *et al.*, 1998; Carreiro and Tripler, 2005; Carreiro *et al.*,

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Chapter 19). Examples include comparisons of remnant forest patches with different land-use histories within an urban area (Hobbs, 1988b; Zipperer, 2002) and between remnants in urban, suburban and rural areas (McDonnell *et al.*, 1997; Pavao-Zuckerman and Coleman, 2005; Carreiro *et al.*, Chapter 19).

Apart from the comparative approach, patch dynamic and watershed approaches have been promoted to study the interaction of the human and ecological domains of the entire urban landscape (Pickett *et al.*, 1997b; Grimm *et al.*, 2000; Grimm and Redman, 2004). Urban landscapes are diverse spatial mosaics representing a variety of ecological conditions that are useful in comparing the effects of urban land-use change on ecological structure and function. Natural sources of spatial heterogeneity in ecosystems still underlie and constrain the effects of land-use and land-cover change. These factors include the geophysical setting, physical environment, biological agents, and processes of disturbance and stress (Pickett and Rogers, 1997). Humans introduce an additional source of heterogeneity by altering landforms and drainage patterns, constructing structures, introducing non-native species, and modifying natural disturbance regimes (Turner *et al.*, 1990; Pouyat *et al.*, 2007a). Additional patchiness also results from variations in human behaviour and social structures that function at different scales (Grove *et al.*, 2006). By comparing human and natural sources of heterogeneity within the spatial context of watersheds, the processes and patterns of human and ecological systems can be linked (Pickett *et al.*, 2001, Chapter 3).

In this chapter we investigate urban biogeochemical cycles from studies conducted in the New York City and Baltimore, Maryland, metropolitan areas. Since the United States is an industrialised country, our approach may not be applicable in cities within developing nations, where urban populations are expanding at a tremendous rate and development patterns differ from those of cities in developed countries (Zipperer and Pickett, 2001). We focus on carbon (C) and nitrogen (N) cycling, both of which have environmental importance from a local to global scale (Vitousek *et al.*, 1997a; Schlesinger and Andrews, 2000). The biogeochemistry of these elements might be a useful indicator of the long-term accrued effects of urban stress and disturbance on ecosystem structure and function (Pouyat *et al.*, 1995a). Here we evaluate: (1) the effects of urban land-use change on native ecosystems; and (2) the relative strengths and limitations of three approaches used to investigate urban land-use effects on C and N cycling: the urban gradient, patch dynamic and watershed approaches.

Urban land-use change

When land is converted from forest, grassland and farmland to urban land use, novel management and disturbance regimes are introduced by humans (Pouyat *et al.*, 2003). Most large-scale disturbances, such as site grading and vegetation removal, occur in the construction phase of urban development, whereas finer-scale disturbances, such as demolition of old buildings and conversion of vacant lots into community gardens, generally occur later. Horticultural management introduces even finer-scale disturbances and includes the establishment and maintenance of lawns, shade trees and planting beds on parcels of land that typically are smaller than the parcels that were managed in the previous forest or agricultural landscape. Horticultural management generally does not result in continuous physical disturbance of soil or plant communities, so it has less impact on biogeochemical cycles than management of agricultural systems, which disturbs plant and soil systems annually or continuously throughout the year (Pouyat *et al.*, 2003).

Urban environmental factors that could affect biogeochemical cycles are overlaid on these novel patterns of disturbance and management. These factors include the urban heat island effect; increased atmospheric concentrations of carbon dioxide (CO_2), and oxides of nitrogen and sulphur; atmospheric N deposition; heavy metal and organic chemical contaminants; and the introduction of invasive plant and animal species (Carreiro *et al.*, Chapter 19). In some metropolitan areas, the net effects of these multiple factors may be analogous to predictions of global environmental change, e.g. increased temperatures and rising atmospheric concentrations of CO_2 (Pickett *et al.*, 2001; Carreiro and Tripler, 2005). In fact, some of these environmental factors seem to affect biogeochemical cycles in remnant forest ecosystems within metropolitan areas (Grodzinski *et al.*, 1984; McDonnell *et al.*, 1997). The results of these and other studies are reviewed by Carreiro *et al.* (Chapter 19).

The net result of human disturbance, landscape management and environmental changes associated with urbanisation is a mosaic of land patches with varying environmental conditions (Pouyat *et al.*, 2003). In this heterogeneous landscape, distinctly different patch types lie near each other, e.g. a forest patch adjacent to a residential area. As landscapes become more densely developed, the structure and function of the managed patches should become more distinctive and correspond more closely to the management preferences of the individual land owner, e.g. whether a parcel is landscaped primarily with native or exotic plants (Zipperer *et al.*, 1997). Typically, the development pattern of a metropolitan area results in a configuration where patch size tends to decrease from rural outlying areas towards the urban core while the diversity tends to be greatest at the urban fringe (sensu Burghardt, 1994) (Fig. 20.1). Management of these parcels depends on ownership (public or private) and/or on the socio-economic status of the owner (e.g. Hope *et al.*, 2003; Law *et al.*, 2004). By contrast, remnant patches such as a forest or wetland patch typically are not managed and will have

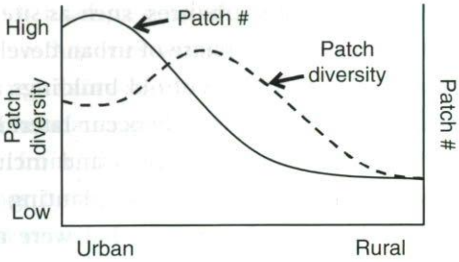

Fig. 20.1. Conceptual relationship of patch diversity and density going from a highly urbanised to rural landscape. Patch density peaks near the urban core while patch diversity peaks at the urban fringe (sensu Burghardt, 1994). Figure used by permission, after J. Russell-Anelli.

characteristics of the native ecosystem, but with significant changes resulting from the effects of the urban environment (Pouyat *et al.*, 1995a; Guntenspergen *et al.*, Chapter 29). In urban landscapes, the immediate surroundings of individual patches are likely to exert a strong influence on biogeochemical cycling, which is due to higher edge-to-interior ratios and thus more open flows of energy, matter and organisms between the patch and adjacent urban matrix (Carreiro *et al.*, Chapter 19).

Based on these observations, it is important to consider and map the arrangement of patches and their relationships with each other in urban landscapes (Forman, 1995). Individual patches can be studied as black boxes with fluxes and cycles of resources that interact with neighbouring patches (Zonneveld, 1989; Grimm *et al.*, 2003). For example, patches within a watershed might function as sources or sinks of nutrients and contaminants and also regulate matter and water flows. Thus, in modelling watersheds hydrologists calculate the inputs of water, nutrients and contaminants to streamflow depending on the cumulative biotic and abiotic attributes of individual patches and their spatial location within a watershed (Black, 1991). In the case of remnant patches such as a forest or wetland, the environmental context of individual patches, whether it be an urban, suburban or rural matrix, can be related to C and N cycling within the patch (Pouyat *et al.*, 1995a).

Urban–rural gradient approach

The gradient approach has been used to investigate effects of urban environmental changes on C and N dynamics in remnant forest patches (McDonnell *et al.*, 1997; Carreiro *et al.*, Chapter 19). The approach is particularly effective as urban environments consist of many factors and effects that otherwise would be difficult to manipulate (Pouyat *et al.*, 1995a). The use of the environmental gradient paradigm to investigate responses of forest ecosystems to urban land-use change was first proposed by McDonnell and Pickett (1990), although the approach had been used earlier by Santas (1986) to study soil organisms. McDonnell and Pickett recognised that complex environmental gradients also may occur when human population densities, human-built structures and human-generated processes vary spatially on a landscape. The environmental gradient paradigm was introduced by Whittaker (1967) as a method for examining and explaining species composition of forest communities along an elevation gradient. Whittaker's assumption was that environmental variation is ordered in space and that the spatial pattern of the environment constrains the structural and functional components of ecosystems (McDonnell and Pickett, 1990). McDonnell and Pickett suggested the term urban–rural land-use gradient to describe environmental gradients in metropolitan areas caused largely by variations in land use (Pickett *et al.*, Chapter 3).

Applying the gradient approach to metropolitan areas

Gradient analysis techniques can either be direct when the underlying environmental factor is ordered linearly in space or indirect when a gradient of underlying factors is organised non-continuously across the landscape (McDonnell *et al.*, 1993). These techniques may require multi-variate statistical approaches when the gradient consists of several environmental factors, some of which may co-vary and interact. For a discussion of the theory of gradient analysis, see Ter Braak and Prentice (1988), and as it is applied to urban–rural gradients, McDonnell *et al.* (1993). Indirect techniques entail measuring ecological system parameters and then ordinating these values to represent the underlying environmental gradient (Whittaker, 1967; Ter Braak and Prentice, 1988). The ordinated response variables, or surrogate variables, are then compared with actual site factors or environmental measurements such as soil type and the availability of soil moisture. These site factors and environmental measurements can then be related to other ecosystem attributes. For example, Brush *et al.* (1980) conducted a gradient analysis of woody vegetation data for the State of Maryland, USA. The authors found a spatial correlation between forest associations (surrogate variables) and soil type (site factors). They concluded that patterns of soil type (soil texture, slope position) were related to water availability, which was ultimately controlling the distribution of woody species in the state.

McDonnell *et al.* (1993) suggested that urban–rural gradients are complex and non-linear, and thus best described by indirect gradient analysis (Pickett *et al.*, Chapter 3). Although measures or indices of urban land cover and land use are easily obtained, often there are few data that describe the spatial variation in environmental factors such as soils that underlie cities and suburbs. As a result, it is difficult to determine how environmental variation interacts with and relates to attributes of the urban ecosystem and diverse human activities. In other words, we do not fully understand how urban environmental factors vary spatially or what causes the spatial pattern – a relationship that is assumed when performing a gradient analysis. Moreover, even if relationships are found, we may not be able to describe the mechanistic links behind a particular response. For example, in describing forest associations in Maryland, Brush *et al.* (1980) understood the relationship between plant species and soil texture, and between soil texture and soil moisture. This was possible because the interrelationship between soil texture and moisture had been well studied, and such soil patterns had been well described at the scale of landscapes and physiographic regions. In urban ecosystems, we lack the knowledge base for making such connections, so before attempts are made to correlate an ecosystem response to a suspected urban environmental gradient, we need to quantify the relationships between human activities, urban features in the landscape and pre-existing (underlying) spatial patterns for environmental variables known to drive the responses of interest.

Landscapes are commonly classified by geographers as urban, suburban or rural/wildland on the basis of socio-economic data and political entities. However, the ecological implications of these classifications remain unclear (McDonnell and Pickett, 1990; Medley *et al.*, 1995; Yang and Zhou, Chapter 16). The following example illustrates this point. Pouyat *et al.* (1995b) measured several soil variables (heavy metals, basic cations, content of soil organic matter, electrical conductivity) in oak forest stands along an urban–rural land-use gradient in the New York metropolitan area. Soil data were submitted to indirect gradient analysis (Principal Component Analysis) in which the scores for the first two principal components of each forest patch were plotted (Fig. 20.2). Using the results of this analysis, we assigned each forest patch to a land-use type (U = urban, SU = suburban and R = rural in Fig. 20.2) based on the criteria of a regional planning agency (Yaro and Hiss, 1996), which used political boundaries (counties) and population density to distinguish urban from suburban and suburban from rural land-use types. Inspection of the groupings with respect to the PC1 axis shows that the suburban and rural forest patches are overlapped, while one of the patches in the urban group is more closely aligned to the suburban patches (Fig. 20.2).

In this example, the qualitative assignment of patches to a land-use type based on planning criteria did not correspond directly to the ordination of plots based on soil criteria, suggesting the need for a more quantitative ecological definition of land use. Indeed, in a review of research using urban–rural land-use gradients,

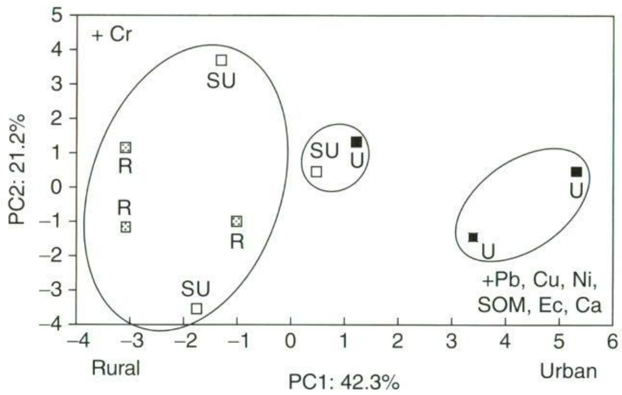

Fig. 20.2. Scatter plot of the first two principal component scores of soil chemical properties measured along an urban–rural land-use gradient in New York metropolitan area. Percentage of variation explained by each component is given on each axis. Each square represents a forest patch ($n = 9$ plots). Land use (U = urban, SU = suburban and R = rural) based on urban planning designations is next to each patch. Solid lines show cluster analysis of quantified geographical features (modified from Pouyat *et al.*, 1995b).

Theobald (2004) identified a lack of quantifiable metrics in the description of human-modified systems in the studies reviewed. For the New York urban–rural land-use gradient, Medley *et al.* (1995) used quantitative metrics such as patch size, traffic volume and population density. In the previous example, the landscape context of each forest patch along the gradient was then quantified by Pouyat *et al.* (1995b) using the Medley measures. Using results of this analysis and for illustrative purposes here, we regrouped the patches (Fig. 20.2, solid lines) and found more clearly defined groupings that corresponded to the ordination of patches better than those defined qualitatively by planners.

Nonetheless, even if a quantitatively defined urban–rural land-use gradient is available, the spatial extent and variation of underlying environmental factors may be poorly understood. To illustrate this point, we again used soil data from the New York urban–rural land-use gradient study. However, this time we used a direct gradient approach to compare relationships between different quantitative measures of urban land use and soil response variables. We first used distance to the urban core (a variable that is easily measured) to serve as a surrogate (and continuous) variable for the extent of urban development and as a predictor of soil characteristics of the urban, suburban and rural forest plots. There was a significant relationship between soil lead (Pb) content and distance from the urban core (Central Park in Manhattan) ($r = 0.677$, $P = 0.05$ using Pearson Correlation). The relationship was even stronger when soil Pb levels were related to traffic volume within an area of 1 km^2 around each stand ($r = 0.953$, $P = 0.001$ using Pearson Correlation). That soil Pb content was correlated more highly with traffic volume than with distance to the urban core suggests that Pb emissions from automobiles may be more responsible than other Pb sources for the variation in soil Pb contamination along this urban–rural gradient. Moreover, traffic volume may not necessarily be strongly related to distance to the urban core. The use of more quantitative measures of urban development revealed a functional relationship among a site environmental variable (soil Pb content), an urban environmental factor (Pb deposition) and its probable source (automobile exhaust).

This example suggests that when possible, urban gradients should be defined quantitatively using both geographical features, e.g. road density or traffic volume, and environmental factors, e.g. heavy metal deposition. The latter can be substituted with site variables that are related to a particular environmental factor, e.g. content of soil heavy metals, as a temporal integrator and index of atmospheric heavy metal deposition (Table 20.1). Further, using distance to the urban core as a surrogate measure for urban land use may not always be appropriate to describe an underlying environmental gradient (McDonnell and Hahs, Chapter 5). Once these relationships are better understood, additional ecological meaning and explanatory power can be derived from statistical relationships between urban land use and the ecosystem response that is measured. A similar conclusion was reached by plant ecologists, who now recommend submitting environmental variables prior to vegetation variables to plant community ordination analyses (Fralish, 2002).

Another challenge in quantifying urban–rural land-use gradients and the underlying environmental factors associated with such gradients is separating the effect of non-urban and urban environmental factors. For example, soil characteristics of oak/tulip-poplar forests in the Baltimore metropolitan area were measured along an urban–rural land-use gradient (Szlavecz *et al.*, 2006). After correlations among the soil variables were explored using Principal Component Analysis, the plots were labelled according to their land-use context (Fig. 20.3). However, whereas canopy cover was similar among plots, soil type varied because of the variability in surface geology of the region. Thus land-use type is confounded with variation in soil type, so it is unclear whether differences in soil characteristics (and C and N processes) measured along the urban–rural land-use gradient were due to urban environmental factors or natural soil-forming factors (Fig. 20.3).

To increase our ability to separate urban from non-urban effects, Pouyat (1991) suggested using the Factor Approach, a conceptual model first proposed by Jenny (1941) to describe the formation of soil at landscape scales. This approach

Table 20.1. *Potential quantifiable metrics that can be used to define urban–rural land-use gradients.*

Urban features	Environmental factors	Site environment variables	Ecosystem variables (C and N)
Distance	**Atmosphere**	**Soil chemical**	**Primary productivity**
Human population			**Decomposition**
	• Wet/dry deposition	• Heavy metals and other contaminants	**N mineralisation**
• Density	• Ozone	• Acidity/alkalinity	**Nitrification**
• Per capita use	• Carbon dioxide	• Hydrophobicity	**Denitrification**
		• Calcium	**Soil respiration**
Human structure	**Climate**	• Phosphorus	**Food web structure**
		• C:N ratios	**Species diversity**
• Road density	• Precipitation		**C pools**
• Impervious (%)	• Air temperature UVB radiation		**N pools**
• Urban land (%)		**Soil physical**	**N retention**
• Housing density	**Biological**	• Temperature	
• Density of birdfeeders	Non-native species propagules	• Moisture	
		• Infiltration	
Human function		• Bulk density	
• Traffic volume			
• Energy use		**Biological**	
• Water use		• Seed bank	
• Pollution emissions		• Number of species	
		• Number of invasive species	
Habitat measures			
• Fragmentation			
• Connectivity			
• Patch size distribution			
• Vegetation structure			
• Disturbance frequency			

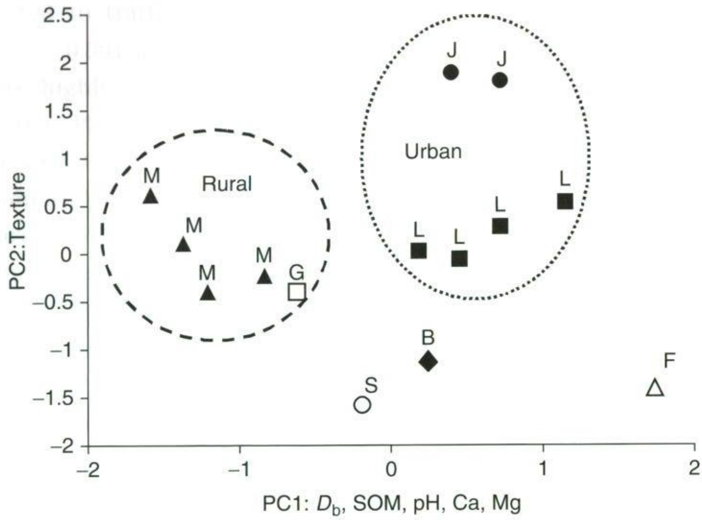

Fig. 20.3. Scatter plots of the first two principal component scores of soil chemical and physical properties measured along an urban–rural land-use gradient in Baltimore metropolitan area. Each symbol represents a forest patch ($n = 3$ plots). Soil type is designated with a capital letter adjacent to each symbol. Forest patches are grouped by either an urban or a rural context (hatched lines). D_b = soil bulk density; SOM = soil organic matter concentration.

posits that soil and ecosystem development is determined by a combination of state factors that include climate (*cl*), organisms (*o*), parent material (*pm*), relief (*r*) and time (*t*), where the characteristics of any given soil (or ecosystem), *S*, are the function $S = f(cl, o, pm, r, t)$. Amundson and Jenny (1991) and Pouyat (1991) proposed that human effects can be incorporated into the factor approach by including a sixth or anthropogenic factor *a*, such that $S = f(a, cl, o, pm, r, t)$. To investigate the relative importance of individual factors, Jenny (1961), Vitousek *et al.* (1983) and Van Cleve *et al.* (1991) identified 'sequences' of soil bodies or ecosystems on landscapes in which a single factor varies while the other factors are held constant, e.g. a chronosequence where age, *t*, is the varying factor. Likewise, Pouyat (1991) and Pouyat and Effland (1999) proposed that urban–rural land-use gradients can create situations in which the anthropogenic factor *a* (in this case, an urban factor) varies for relatively short distances (km) so that it is possible to hold the remaining factors as constant as possible, i.e. an 'anthroposequence', where $S = f(a)_{cl,o,pm,r,t}$. With the Factor Approach, the null hypothesis that there is no detectable difference in the response variable along the urban–rural gradient is essentially predetermined (Pouyat *et al.*, 1995b). Further, a study design that emulates an anthroposequence should a priori account for the potential of confounding urban (*a*) and non-urban (*cl*, *o*, *pm*, *r*, *t*) factor effects.

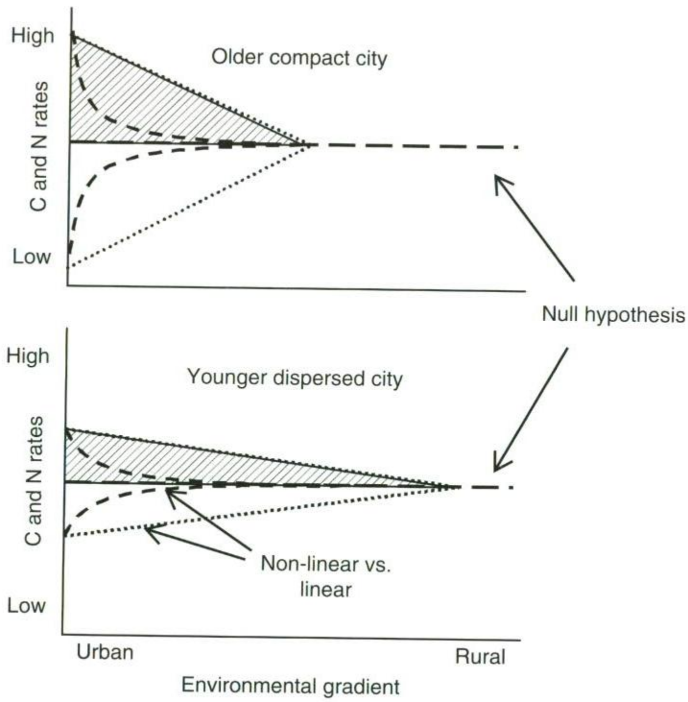

Fig. 20.4. Conceptual diagrams of C and N rate responses to an urban–rural environmental gradient for an old, compact city (top) and young, dispersed city (bottom). Hatched lines under response curves represent stimulation of C and N rates. Null hypothesis is represented by horizontal dashed line, i.e. no change in rates.

Other considerations related to the urban gradient approach

When using the gradient approach to study urban environmental effects on remnant patches, it is important to consider the physical, biological and socio-economic characteristics of the metropolitan area under investigation (Niemelä *et al.*, Chapter 2, Pickett *et al.*, Chapter 3). Consider a suite of potential response curves of C and N measurements in forest patches situated along an urban–rural land-use gradient (Fig 20.4). As stated earlier, the underlying environmental gradient consists of many factors that can affect C and N cycling. These factors are themselves related to the development pattern of the city and its surrounding area, for example population density (effects also separately dependent on aerial extent of the city and total population), dominant commercial activities and age of the city (see McDonnell and Hahs, Chapter 5). In older more compact and industrialised cities, the differences in atmospheric pollution between urban and rural ends of the gradient should be steeper and less linear (with a more abrupt threshold effect evident) than in newer less industrialised cities that have a dispersed development pattern (Fig. 20.4).

Table 20.2. *Comparison of trends in soil characteristics and C and N process rates along urban–rural land-use gradients in three cities that range in population from over 61 607 (Asheville, NC) to 7 420 166 (New York City, NY) inhabitants.*

Population	New York City *7 420 166*		Baltimore *645 593*		Asheville *61 607*	
Soil variable	Rural	Urban	Rural	Urban	Rural	Urban
pH	4.7	4.5	4.6	5.2	4.9	4.9
SOM ($g\,kg^{-1}$)	75	108	110	90	97	79
Annual temperature (°C)	8.5	12.5	12.8	14.5	11.9	13.0
N-mineralisation ($mg\,N\,kg^{-1}\,SDW\,d^{-1}$)	4.02	10.3	2.2	8.0	0.11	0.26
Leaf decay ($mg\,d^{-1}$)	0.0068	0.0113	n.a.	n.a.	0.0012	0.0009

Note:

Modified from Pavao-Zuckerman and Coleman, 2005.

In some cases, both the direction and the magnitude of a variable or biogeochemical process can differ along urban–rural land-use gradients in different cities (see litter decay rate, soil pH and organic matter content in Table 20.2 and Carreiro *et al.*, Chapter 19). These results suggest that the net effect of urban environmental factors can stimulate or suppress C and N process rates (Fig. 20.4). In the former case, factors such as N deposition, CO_2 enrichment and temperature increases can increase C and N processing when these factors limit biological activity, which is true for most terrestrial ecosystems. In the latter case, environmental factors such as heavy metals and ozone might constrain biological processes gradually as contamination increases or rapidly after reaching a certain threshold (Fig. 20.4). In either case, the age, dominant commerce and industry, and developmental pattern of the metropolitan area affects the amount of contamination or inputs of C and N and, therefore, the response of C and N processing along an urban environmental gradient (Carreiro *et al.*, Chapter 19).

Another consideration in using the gradient paradigm is the difficulty encountered in determining the underlying mechanisms when a property is correlated with a particular environmental gradient (Vitousek and Matson, 1991). Because many factors underlie an existing environmental gradient, the gradient paradigm can only suggest possible explanations, and the patterns within a particular system may not be valid for other systems (Duarte, 1991). Obviously, the inability of the gradient approach to attribute cause to a gradient correlate is a disadvantage, although one can identify the relative importance of environmental factors and the measured range in which those factors occur in the field. Once identified, the effect of individual factors can be verified and the mechanism of the response can be tested in field and laboratory experiments.

Assessing relationships between environmental factors and ecosystem responses along urban gradients is clearly a complex task. However, not only are these relationships difficult to uncover but they can also lead to erroneous conclusions if field observations are not combined with experimental approaches. Again, we use the New York urban gradient study as an example. In measuring the decay of red oak leaf litter in laboratory incubations, Carreiro *et al.* (1999) found that litter collected in urban forest remnants decayed more slowly than suburban litter, which, in turn, decayed more slowly than rural litter. The authors attributed the differences in decay to intraspecific differences in leaf-litter quality; the urban litter had the poorest quality. Later, oak litter incubating in forest remnants of origin showed that decay rates did not differ statistically across the gradient even though urban litter quality was demonstrably lower in the laboratory assays (Pouyat and Carreiro, 2003). A simple manipulation – exchanging bags of rural and urban litter – showed that warmer temperatures and the presence of invasive earthworm species in the urban plots were likely to compensate for lower quality of urban litter (Pouyat and Carreiro, 2003). The use of field and laboratory studies in combination provided greater explanatory power than otherwise would have been possible.

Finally, the previous example suggests the importance of invasive species in the processing of C and N in ecosystems. Indeed, urban land-use change can modify native habitats and thus elevate indigenous species extinction rates while increasing invasions of non-native species (McKinney, 2002). As a result, the combination of these effects has created a pattern in which native species richness decreases from outlying rural areas to urban centres while invasive species richness increases (Blair, 2001). Since invasive species can play a disproportionate role in controlling C and N cycles in terrestrial ecosystems (see Ehrenfeld, 2003; Bohlen *et al.*, 2004), the relationship between invasive species abundances and habitat change along urban–rural land-use gradients has important implications for the biogeochemical cycling of C and N.

Patch dynamic and watershed approaches

In non-urban ecosystems, spatial heterogeneity can help or hinder the flow of materials and energy across boundaries, affecting biogeochemical cycles within particular patches and across the entire landscape (Gosz, 1991). Patch dynamics treats spatial heterogeneity at any scale as a mosaic whose elements and overall configuration can shift through time. This concept is an important organising principle in ecology and has contributed to the elucidation of the role of spatial control in ecological systems (Wu and Loucks, 1995; Cadenasso *et al.*, 2006). For example, patch dynamics has been cited in explanations for long-term variations in nutrient cycling in forested ecosystems (Likens and Bormann, 1995), and, more recently, in the biogeochemical cycling of nutrients and contaminants in urban landscapes (Pickett *et al.*, 1997b; Grimm *et al.*, 2000; Pickett *et al.*, Chapter 3).

How patches function and interrelate are important questions in urban ecology (Grimm and Redman, 2004). Urban landscapes are more complex than natural landscapes and the importance of human activities makes it difficult to predict their interactive effects on ecosystem processes and structure. If we are to understand biogeochemical cycles in urban ecosystems we must integrate human behaviour and socio-economic factors with ecological factors. Although social heterogeneity is not addressed in this chapter, Pickett *et al.* (1997b; 2001) discussed this topic in detail. They concluded that in urban landscapes, the heterogeneity of both social and natural components can be organised hierarchically around drainage patterns. Thus, the watershed approach is an important tool in assessing the interaction of these components.

Baltimore LTER: using landscape heterogeneity as an asset

Environmental patchiness can be studied at various spatial scales (Wu and Loucks, 1995). In the Baltimore Long-Term Ecological Research (LTER) site, a hierarchical study design is being used to test the hypothesis that ecological and socio-economic heterogeneity operating at different scales affects biogeochemical cycles in urban ecosystems (Pickett *et al.*, Chapter 3). In this study, patches were delineated by their land use and cover and were organised using a nested hierarchy of increasingly larger hydrologic units and watersheds (Pickett *et al.*, 1997b, 2001). Each watershed and their arrangement of different patches were used as whole-ecosystem studies to compare within-patch C and N pools and fluxes among different patch and watershed units at different scales (Table 20.3).

To establish a hierarchical study design, a network of watershed monitoring stations was located within the 17 150-ha Gwynns Falls watershed. The network traversed a gradient in land use from the highly urban core of Baltimore City, through older high-density residential areas, to medium-density single attached houses in the middle reaches, and finally to the rapidly urbanising headwaters of Baltimore County (Doheny, 1999). As part of the network, longitudinal main channel sites were established that represented different land-use boundary zones along the Gwynns Falls. The smallest watersheds selected for monitoring in the network (<100 ha) were predominately of a single land-use type, e.g. high- or low-density residential, forest, or agriculture. Thus, differences in socio-economic factors, land use and cover among the smaller watersheds in

Table 20.3. *Nested hierarchical study design of the Baltimore LTER. Patches are delineated and classified by their land use, cover, built structures, and land management activities and organised using a nested hierarchy of increasingly larger hydrologic units and watersheds.*

Hydrologic hierarchy	Size in area (ha)	Classification system	Minimum patch size	Patch comparisons
Regional watershed	>50 000	Anderson II	15 ha	Land use/cover
Watershed	17 125	Anderson III, HERCULES[a]	4 ha	Land use/cover Vegetation, built structures and surfaces
Sub watershed	100–2500	Anderson III, HERCULES	0.5–4 ha	Land use/cover Vegetation, built structures and surfaces
Small watershed	<100	HERCULES, Ecotope Level I[b]	0.5 ha 25–100 m^2	Vegetation, built structures and surfaces Landscape features, cover/management
Neighbourhood catchment	<10	Ecotope Level I	25–100 m^2	Landscape features, cover/management
Stream reach/ hillslope, city block	<1	Ecotope Level II[b]	8 m^2	Land management, cover

Notes:

[a] Cadenasso *et al.* (2006).

[b] Ellis *et al.* (2006).

the hierarchy were used to set up comparisons much like the gradient analysis of remnant forest patches described earlier. These comparisons or 'natural experiments' substitute for the large-scale manipulations that have been used on small watersheds at other LTER sites (Bormann and Likens, 1979; Hornbeck and Swank, 1992). As a result, the monitoring of small watersheds has allowed comparisons with other LTER sites and more detailed analysis of input–output mass balance than those of larger watersheds of mixed uses (Groffman *et al.*, 2004).

Unlike the more homogeneous small watersheds, larger watersheds (>100 ha) in the hierarchy have mixed uses, so it is more difficult to connect outflows and nutrient loads to a particular land-use or patch type. For these watersheds, patch structure and function has been investigated using the variable source area approach to model watershed hydrology and nutrient dynamics (Band *et al.*, 2000; 2001). The net effect of patches of varying composition, management regimes and site histories can be estimated with variable source area approaches by modelling how the attributes of different patches cycle or contribute water, nutrients and contaminants depending on their location in the watershed (Black, 1991). Moreover, sensitivity analyses can be conducted in different modelling runs to determine how varying patch compositions and configurations affect quantitative and qualitative inputs to streams.

In addition to modelling, a network of 'intensive' and 'extensive' plots was established in representative patch types to capture the range of spatial and temporal conditions within each watershed in the hierarchy. The goals were to measure ecosystem response variables over time and conduct whole-ecosystem analyses in a representative patch type. The intensive plot measurements are important for calibrating models, developing mass-balance budgets and measuring ecosystem responses to stochastic events and climate fluctuations (Groffman *et al.*, 2006). Intensive plots require a high commitment of resources and time for sampling, so only a limited number have been established. By contrast, larger numbers of extensive plots are sampled intermittently to assess spatial variation in several ecosystem response variables in the metropolitan region, e.g. plant productivity (Nowak *et al.*, 2003) and soil chemistry (Pouyat *et al.*, 2007b).

Developing functional characterisations of patch structure that are ecologically based is important for identifying representative plots in different patch types and for aggregating C and N mass-balance measurements for a specified watershed or other ecologically bounded area. Until now, the location of extensive plots has been stratified according to Anderson land-use categories (Anderson *et al.*, 1976). However, new land-cover classification systems with higher categorical resolution are being developed to improve the stratification of plots primarily for highly heterogonous urban mosaics (Ellis *et al.*, 2006; Cadenasso *et al.*, 2007; Table 20.3).

Patch comparisons

The nested hierarchical design has allowed comparisons of C and N cycling processes of different patch types at different spatial and temporal scales. Measurements of trace-gas fluxes and nitrogen-cycling variables on our intensive plots revealed temporal variation in natural processes as well as spatial variation caused by land-use change. For example, *in situ* measurements of net N mineralisation and nitrification on our intensive forest plots showed that the magnitude and annual variation of natural fluxes are much higher than many anthropogenic fluxes. Net nitrification (a natural source of nitrate) ranged from approximately 5 to 15 $\mathrm{kg\,N\,ha^{-1}\,yr^{-1}}$, equal in magnitude to atmospheric deposition and fertilisation fluxes in our watersheds (Table 20.4). Comparison of

Table 20.4. *Nitrogen budget input–output analysis (kg/ha per year) for a suburban and a forested watershed (<100 ha) in the Baltimore LTER study.*

	Suburban	Forest
Inputs		
Atmosphere	11.2	11.2
Fertiliser	14.4	0
Total	25.6	11.2
Outputs		
Stream flow	6.5	0.52
Retention		
Mass	19.1	10.7
Percentage	75	95

Note:
Adapted from Groffman *et al.* (2004).

grass and forest intensive plots has shown that grass areas have surprisingly high N retention and moderate leaching losses, which are likely to be due to an active carbon cycle (indexed by high total soil respiration) maintained by the young, actively growing grass (Pouyat *et al.*, 2007a).

Unlike the intensive plots, results from the extensive plots provide one-time measures of soil characteristics, C and N cycling, and vegetation structure. However, using a hierarchical classification system to stratify plots made it possible to compare among and within patch classes at different scales. For example, a network of 200 extensive plots was stratified by the relatively coarse scale Anderson Level II land-use and cover classes within Baltimore City (Anderson *et al.*, 1976; Table 20.3). The plots were sampled for vegetation structure and soil over 2 years (Nowak *et al.*, 2003; Pouyat *et al.*, 2007b). There was a wide range in soil characteristics among all plots, although a subset of the variables measured (P, K, bulk density and pH) showed a discernible and coherent pattern with Anderson land-use classes. Differences were greatest between land-use types characterised by intensive land management (lawns) and the absence of management (forests). In particular, concentrations of P and K, which are in most lawn fertilisers, differentiated the most between forest and grass cover (Pouyat *et al.*, 2007b).

In separate studies, N-transformation rates were measured on a subset of the extensive plots. For illustrative purposes, we compare potential nitrification rates among Anderson land-use classes (P. Groffman, C. Williams, R. Pouyat and I. Yesilonis, unpublished data) with results from remnant forest patches along an urban gradient in the Baltimore metropolitan area (Szlavecz *et al.*, 2006).

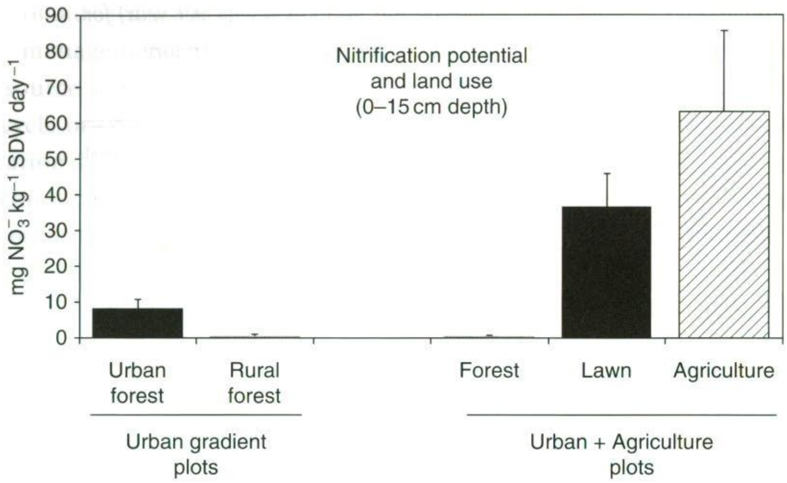

Fig. 20.5. Mean (± standard error) of potential net nitrification rates (mg NO_3^- kg^{-1} day^{-1}) of mineral soil samples of forest, lawn and agricultural plots in the Baltimore LTER study. Bars on left represent comparison of urban ($n = 9$) and rural ($n = 9$) forest patches. Bars on right represent comparison of forest, lawn and agriculture land-use types ($n = 14$, 10 and 10, respectively). SDW = soil dry weight. Data from P. Groffman, C. Williams, R. Pouyat and I. Yesilonis (unpublished) and Szlavecz *et al.* (2006).

Differences were much higher among land-use types than between urban and rural forest patches (Fig. 20.5). These results suggest that soil management associated with different land uses has a much greater effect on C and N cycling than abiotic environmental variables that are altered in the Baltimore metropolitan area, such as the heat island effect and deposition of atmospheric pollutants.

Because of the wide distribution of a relatively large number of plots, it was possible to reclassify the extensive plots using criteria other than Anderson land-use classes. For the soil results cited, plots were reclassified by surface geology and parent material since they should have an important influence on initial soil element contents. The reclassification revealed that a subset of the variables measured, primarily Al, Mg, V, Mn, Fe, Ni and soil texture, were strongly related to the presence of a specific rock type in the region (Pouyat *et al.*, 2007b). These results suggest that natural soil-forming factors, parent material in this case, were more important than Anderson land-use classes in determining the spatial pattern of these elements in the Baltimore City landscape. However, at this coarse scale, the variation of some factors, e.g. Pb, was not explained by land use or surface geology, suggesting that spatial variation of certain characteristics is controlled by other factors operating at finer scales.

In using the extensive network of plots, it was possible to separate natural from urban influences on spatial variation of factors that determine rates of

Table 20.5. *Carbon storage and sequestration (±SE) in aboveground biomass of Baltimore City.*

Land use	Storage (t C)	Storage (t C ha^{-1})	Sequestration (t C yr^{-1})
Forest	124 576 (25 250)	73 (15)	3009 (489)
Medium density residential	139 129 (27 272)	33 (7)	4195 (653)
High density residential	119 321 (31 811)	20 (5)	3423 (670)
Urban open	89 992 (29 734)	60 (20)	2052 (670)
Commercial/industrial	63 665 (34 625)	13 (7)	1862 (1204)
Institutional	29 223 (25 168)	16 (14)	814 (655)
Transportation	4792 (3931)	8 (7)	170 (121)
Barren	83 (82)	1 (0.4)	5 (5)
Total city	570 781	224	15 529

Source: Calculated using the UFORE model (Nowak and Crane, 2000). Data used to run UFORE model from Nowak *et al.* (2003).

C and N cycling processes and other soil characteristics, at least at the scale of the classification categories used. The designation of patches into specific classes within a hierarchical system also allowed like classes to be aggregated into a watershed or other bounded area, whether ecologically or politically defined. For example, vegetation data collected in the extensive plots were used to estimate the amount of C stored (± standard error) in aboveground biomass for Baltimore City using the UFORE model (Nowak and Crane, 2000). UFORE calculates the amount of C stored at the plot scale and then aggregates to the city or regional scale using a hierarchical classification system (Table 20.5). In a similar fashion, belowground storage of soil organic C was estimated using the extensive plot data (Pouyat *et al.*, 2006).

We also can delineate patches with higher categorical resolution and make comparisons at the finer scale of a neighbourhood or small watershed (<100 ha). At this scale, measurements can be related to patches with specific site histories and activities of individual land managers (Pouyat *et al.*, 2007a). In turn these can be compared with ecosystem-level measurements such as small watershed hydrological outflows like those mentioned previously, or above-canopy CO_2 fluxes from a tower. In two suburban neighbourhoods in the Baltimore metropolitan area, patches have been delineated using high-resolution ecotope mapping (Ellis *et al.*, 2000; Ellis, 2004). Preliminary results suggest that socio-economic factors, lot size, and age of the housing development are important explanatory variables for soil variables at this scale (Law *et al.*, 2004; Fig. 20.6).

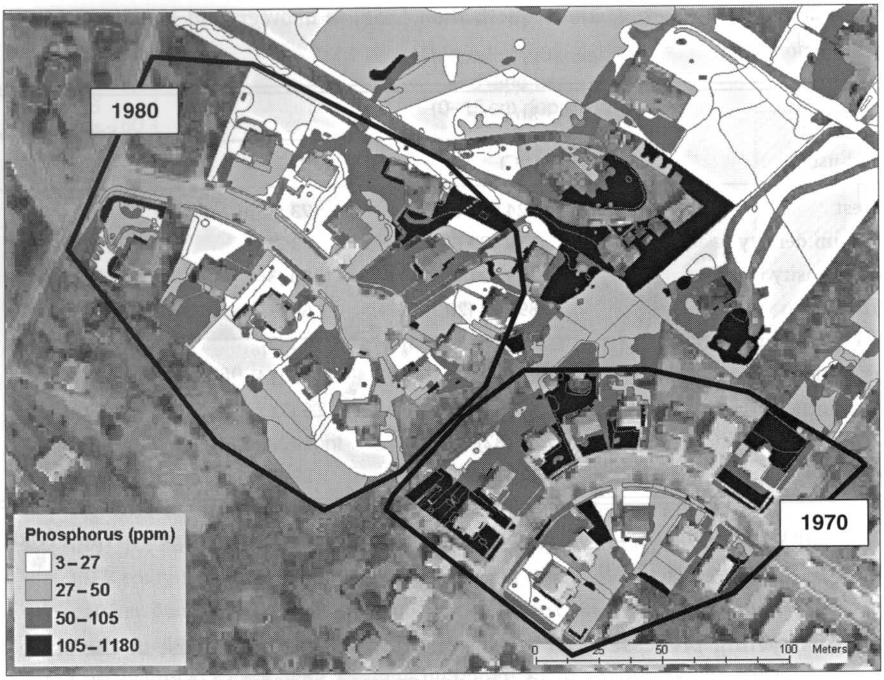

Fig. 20.6. Map of phosphorus (P) concentration ranges for two housing developments in the Cub Hill neighbourhood of Baltimore County. Developments differ by age (1980 upper left and 1970 lower right). Concentrations of P in lawns were significantly higher ($P < 0.01$) in the 1970 ($n = 11$) than the 1980 ($n = 13$) subdivision (J. Russell-Anelli, I. Yesilonis and R. Pouyat, unpublished data).

Whole-ecosystem comparisons

Our long-term watershed monitoring has enabled us to compare outflows of small watersheds dominated by different land uses. Results show that N exports are higher from urban watersheds than from forest watersheds, but lower from agriculture watersheds (Groffman *et al.*, 2004). That these exports were not markedly variable was surprising given the high spatial heterogeneity of these watersheds. Also, our low-density residential watershed (Baismans Run, >100 ha) is served by septic systems and has relatively high nitrate concentrations. This was unexpected given that nearly 75% of its area is forested and our 'reference' forested watershed (Pond Branch) is nested within it and had the lowest concentrations of our monitored watersheds (Fig. 20.7). These results suggest that septic systems can add high concentrations of nitrate directly to groundwater (Gold *et al.*, 1990). The nitrate in our other residential watersheds is likely to have originated from contamination from sanitary sewer systems because the households in these watersheds always have been connected to the sanitary sewer system.

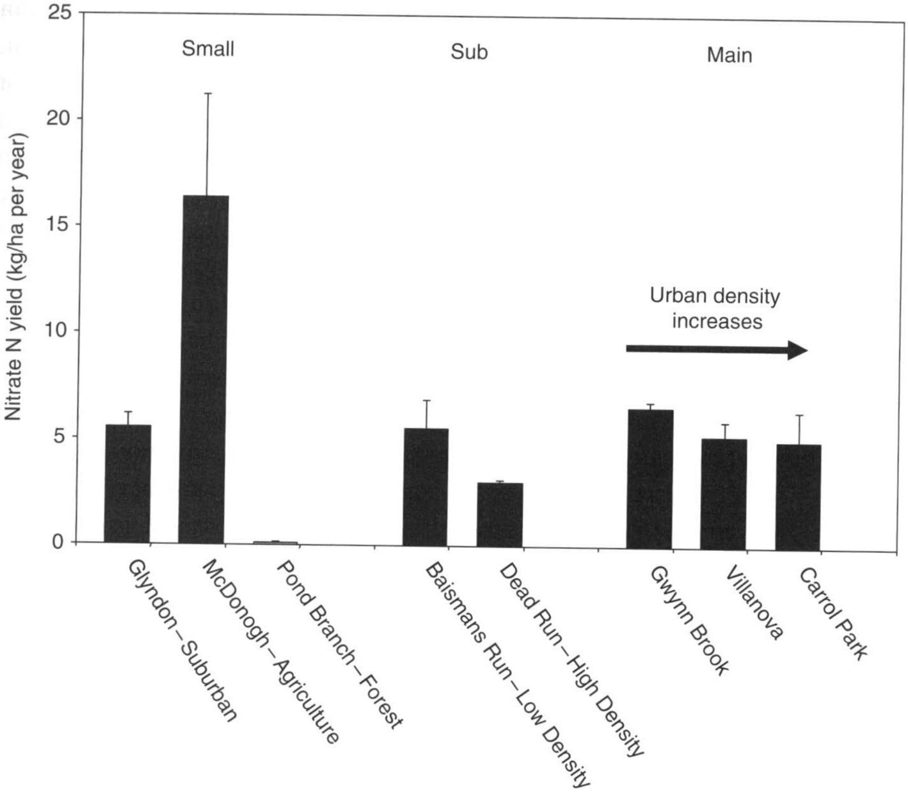

Fig. 20.7. Mean ($\pm$ standard error) nitrate N yields (kg per ha per year) for three water years of small, sub, and main channel catchments of the Gwynns Falls watershed. Small catchments are dominated by suburban (Glyndon), agriculture (McDonogh) and forested (Pond Branch) land uses. Sub and main channel catchments have mixed land uses. Modified from Groffman *et al.* (2004).

The watershed mass-balance approach also allowed us to calculate the retention of N in individual watersheds. For example, N budget input–output analysis showed that retention of N in a suburban watershed (<100 ha) was surprisingly high compared with our reference forested watershed (Table 20.4). These analyses have raised questions about unique sources and sinks for N in urban and suburban watersheds, including residential lawns, leaky sewers, riparian zones, stream features such as organic debris dams, and stormwater detention basins (Groffman and Crawford, 2003; Groffman *et al.*, 2004), which we are addressing in ongoing research.

Concluding remarks

To date, our use of the gradient, patch dynamic and watershed approaches in studies of C and N cycling in urban ecosystems has produced useful and informative results, but additional research is needed before we can integrate these cycles with the behaviour of humans and the built environment. Moreover, there are only a few comprehensive, whole-ecosystem analyses of urban ecosystems worldwide from which comparisons can be made (Grimm *et al.*, 2000; Heinz Centre, 2002). As a result, we do not know whether we can generalise from these few comprehensive studies, especially with respect to the array of urban-development patterns, cultural differences and economies of cities around the world (McDonnell and Hahs, Chapter 5). The number of urban study sites worldwide needs to be greatly expanded to include the entire range of cities and human settlements, regardless of the approach taken, although we suggest combining all three approaches where possible.

A good example of a cross-system comparison is GLOBENET (Global Network for Monitoring Landscape Change), which compares carabid beetle populations along urban–rural land-use gradients throughout the world (Niemelä *et al.*, 2002; Chapter 2). Comparisons of urban–rural gradients among a suite of metropolitan areas allows for region-by-region assessments of changes in the composition of native species, the importance of specific urban environmental factors, and the net effect (native versus urban ecosystem) of these changes. These comparisons can be enhanced by adopting standardised methods (e.g. Robertson *et al.*, 1999; Niemelä *et al.*, 2002), quantifying the environmental gradient of each metropolitan area (e.g. Table 20.1) and 'normalising' the response data to calculate the environmental effect of urban land-use change for a particular gradient analysis (sensu Seastedt, 1984; Tian, 1998).

Similar to comparisons of urban–rural land-use gradient studies, comparisons of budgets of C and N of small watersheds across different cities have been useful. Of particular interest is the retention of N, which has generated a series of studies of unique urban N sources (leaky sewers, lawns, septic systems, degraded riparian zones) and sinks (in-stream processes, lawns, stormwater detention basins). We suggest a long-term iterative process using monitoring, modelling and experimental approaches to increase our understanding of complex urban watershed ecosystems (Carpenter, 1998; McCarthy, Chapter 7). In so doing, we will continue to monitor our streams, develop models and take advantage of natural climatic variation and human actions, e.g. a major upgrade of the sanitary sewer system in Baltimore.

By combining gradient and patch dynamic approaches with the existing watershed approach to ecosystem analysis, we have been able to take advantage of the inherent heterogeneity of urban landscapes to answer questions related to the effects of land-use change on biogeochemical cycles. The combined use of these approaches also has enabled us to conduct natural experiments that can substitute for large-scale manipulations such as the deforestation manipulations accomplished at the Hubbard Brook LTER. These approaches allow integration with the human socio-economic domain. With increasing populations of people living in urban areas worldwide, and with more cities continuing to expand in number and size, the importance of studies aimed at understanding the global and local ecological aspects of urban ecosystems will continue to increase.